SOIL–SOILN Simulations of Water Drainage and Nitrate Nitrogen Transport from Soil Core Lysimeters

J. D. Jabro*, W. L. Stout, S. L. Fales, and R. H. Fox

ABSTRACT

Water resources protection from nitrate nitrogen (NO_3–N) contamination is an important public concern and a major national environmental issue. The abilities of the SOIL–SOILN model to simulate water drainage and nitrate N fluxes from orchardgrass (*Dactylis glomerata* L.) were evaluated using data from a 3-yr field experiment. The soil is classified as a Hagerstown silt loam soil (fine, mixed, semiactive, mesic Typic Hapludalf). Nitrate losses below the 1-m depth from N-fertilized grazed orchardgrass were measured with intact soil core lysimeters. Five N-fertilizer treatments consisted of a control, urine application in the spring, urine application in the summer, urine application in the fall, and feces application in the summer. The SOIL–SOILN models were evaluated using water drainage and nitrate flux data for 1993–1994, 1994–1995, and 1995–1996. The N rate constants from a similar experiment with inorganic fertilizer and manure treatments under corn (*Zea mays* L.) were used to evaluate the SOILN model under orchardgrass sod. Results indicated that the SOIL model accurately simulated water drainage for all three years. The SOILN model adequately predicted nitrate losses for three urine treatments in each year and a control treatment in 1994–1995. However, it failed to produce accurate simulations for two control treatments in 1993–1994 and 1995–1996, and feces treatments in all three years. The inaccuracy in the simulation results for the control and feces treatments seems to be related to an inadequate modeling of N transformation processes. In general, the results demonstrate the potential of the SOILN model to predict NO_3–N fluxes under pasture conditions using N transformation rate constants determined through the calibration process from corn fields on similar soils.

Nitrate nitrogen (NO_3–N) levels in ground and surface waters of agricultural lands have increased over the past four decades as a result of increases in the use of fertilizers and manure. Regions with well-drained soils and high nitrogen inputs have the highest nitrate levels in the water supply. Contamination of drinking water with nitrate from grazed pasture systems as a result of the high concentration of nitrogen (N) returned to the soil in the excreta of grazing animals has been a major environmental concern and possible human health risk in many parts of the world (Ball and Ryden, 1984; Haigh and White, 1986; Steenvoorden et al., 1986; Roberts, 1987; Owens et al., 1992, 1999). These preceding pasture studies have reported that NO_3–N levels in ground water are often greater that 10 mg L^{-1} (above the maximum USEPA contamination limit for public drinking water) when more than than 100 kg N ha^{-1} yr^{-1} is applied to grazed grasslands.

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More recently, water quality computer models have been used to predict the risk of contamination of water resources by agricultural chemicals from cropping systems. However, only a few efforts have been directed toward testing and evaluating these models for grazed pasture conditions (Jabro et al., 1997, 1998; Mohtar et al., 1997).

These field-scale chemical transport computer models are considered useful tools to predict the risk of agricultural chemicals to contaminate surface and ground waters. The models need to be tested and evaluated for the conditions under which they will be used. If these models are accurately evaluated, they will be useful tools in decision-making, predicting environmental problems, and replacing time-consuming traditional experiments.

The number of nonpoint-source agricultural models used to predict nitrate losses through the root zone and beyond has grown rapidly. Examples of numerous existing models include GLEAMS (Leonard et al., 1987), NLEAP (Shaffer et al., 1991), SOIL–SOILN (Bergstrom et al., 1991; Jansson, 1991; Eckerson et al., 1996), LEACHM (Hutson and Wagenet, 1992), RZWQM (USDA Agricultural Research Service, 1992), and GRASIM (Mohtar et al., 1997). Evaluation of these models under various cropping systems, soils, and weather conditions has also received increasing attention over the past decade (Pennell et al., 1990; Khakural and Robert, 1993; Jabro et al., 1998, 1999).

In this work, the accuracy of Version 9.1 of the SOIL–SOILN (Eckerson et al., 1996) model was evaluated by comparison of field-measured and simulated results of water drainage flux and NO_3–N transport from an experiment conducted on a Hagerstown silt loam soil. Three years of field data were collected from intact soil core lysimeters where urine or feces were applied using animal deposition criteria to fertilized orchardgrass sod.

MATERIALS AND METHODS

Leaching Experiment and Treatments

The soil used in the study is classified as a Hagerstown silt loam developed in limestone residuum parent material. Selected data on soil physical and hydraulic properties used in model simulations are listed in Table 1.

The nitrate leaching experiment was initiated in fall 1992 to measure NO_3–N fluxes from N-fertilized orchardgrass pasture using intact soil core lysimeters (Moyer et al., 1996). Twenty-five intact soil core lysimeters (0.6 m in diameter by 1 m long) were installed in fall 1992 adjacent to plots (Fig. 1). The lysimeters were designed, constructed, and installed similarly to those described by Moyer et al. (1996). Water was pumped from the underground carboys approximately two to four times per month, except for summer dry periods, when no water collections were needed, or during winter, when the soil was frozen. Total water amounts were then measured and

Table 1. Physical characteristics for a Hagerstown silt loam soil as used in SOIL model simulations.

	Depth (m)				
Soil property†	0–0.2	0.2–0.4	0.4–0.6	0.6–0.8	0.8–1
Bulk density (Mg m^{-3})	1.36	1.48	1.53	1.67	1.69
Particle size (%)					
Sand	12.2	11.1	12.3	12.4	12.2
Silt	59.9	50.1	45.0	43.3	43.3
Clay	27.9	38.8	42.7	44.3	44.5
Water content (m^3 m^{-3}) at pressures (MPa):					
0.01	0.41	0.37	0.36	0.35	0.35
0.03	0.38	0.35	0.35	0.34	0.33
0.1	0.34	0.33	0.33	0.32	0.30
0.5	0.24	0.29	0.28	0.27	0.26
1.5	0.22	0.26	0.25	0.24	0.24

† Each value is a mean of three observations.

samples were analyzed for NO_3–N using an automated Cd reduction method (USEPA, 1979). More details regarding lysimeters are given in Moyer et al. (1996) and Jabro et al. (1997).

The N fertilizer treatments consisted of a control, a urine application in the spring, a urine application in the summer, a urine application in the fall, and a feces application in the summer. Each of these five treatments also received 280 kg N ha^{-1} as ammonium nitrate split into five equal applications in 1993. In 1994, each treatment received 220 kg N ha^{-1} as ammonium nitrate (168 kg N ha^{-1} divided into three equal applications were applied between 11 April and 16 May; 52 kg N ha^{-1} divided into two equal applications were applied on 16 June and 16 August). In 1995, each treatment received 194 kg N ha^{-1} as ammonium nitrate (86 kg N ha^{-1} were applied on 11 April; 56 kg N ha^{-1} were applied on 1 May; 52 kg N ha^{-1} divided into two equal applications were applied between 9 June and 12 July). Dates and N rates from urine and feces applications are given in Table 2a. The application rates were selected to assure that N was not a limiting factor for plant growth (Noller and Rhykerd, 1974).

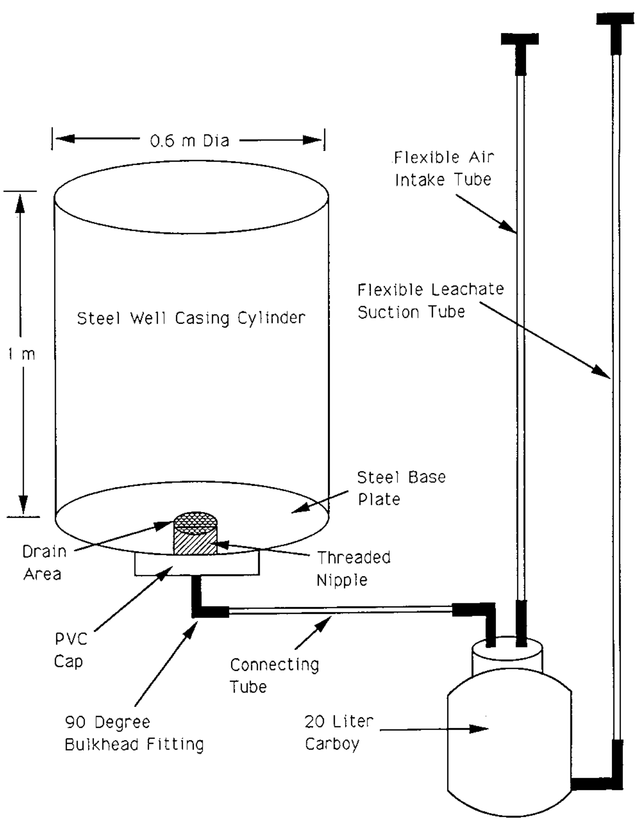

Fig. 1. Schematic diagram of intact soil core lysimeter (adapted from Moyer et al., 1996 and Jabro et al., 1997).

Table 2a. Dates and N rates from urine and feces applications.

	Urine, spring	Urine, summer	Urine, fall	Feces, summer
		1993		
Date	18 May	8 July	21 Sept.	8 July
Rate (g N m^{-2})	96.6	69.9	107.9	28.9
Rate (kg N ha^{-1})	966	699	1079	289
		1994		
Date	28 April	15 Aug.	2 Nov.	15 Aug.
Rate (g N m^{-2})	112	81.2	103.8	28.9
Rate (kg N ha^{-1})	1120	812	1038	289
		1995		
Date	1 May	12 July	31 Oct.	12 July
Rate (g N m^{-2})	141.8	95.6	88.4	27.3
Rate (kg N ha^{-1})	1418	956	884	273

The feces and urine used for the lysimeters were collected from lactating Holstein dairy cows grazing the pasture. The chemical analyses of urine and feces are reported in Table 2b. Each lysimeter urine treatment received 3 L and each feces treatment received 2 kg, the average amounts produced by mature cows per excretion (Petersen et al., 1956). Animal urine and feces were collected on the same day as, or one or two days prior to, lysimeter applications. In cases where collections were not made on the same day as application, the urine and/or feces were stored at 4°C until the day of application. Applications were made in a manner that simulated animal deposition (i.e., merely dropped or poured near the center of the lysimeter surface).

All treatments were replicated five times. Two lysimeters were eliminated, one under urine application in the spring and the other under urine application in the summer, because they did not produce any leachate during the course of the study. Presumably, these two lysimeters did not function properly due to a leak in the connecting tubes.

Plots were harvested seven times during the growing season in both 1993 and 1994 with 2- to 6-wk intervals between harvests. In the 1995 growing season, plots were harvested five times with 4- to 6-wk intervals between harvests. The grass within the lysimeter was clipped manually to a height of 70 to 100 mm and removed in conjunction with grazing of the paddocks to determine the extent of plant N uptake and forage dry matter (Jabro et al., 1997, 1998).

Model Description

SOIL and SOILN are coupled mathematical field-scale models that simulate water and heat transport, N dynamics, and biomass production in a layered soil (Johnsson et al., 1987; Eckersten and Jansson, 1991; Jansson, 1991). The SOIL model predicts water and heat flow between layers in a one-dimensional soil profile. Standard weather data, soil physical and hydraulic properties, and plant characteristics are used as

Table 2b. Chemical analysis of urine and feces.

Material	N	P	K	Organic matter
	% on wet weight basis			
Urine, spring	0.94	0.002	1.00	–
Urine, summer	0.79	0.006	0.74	–
Urine, fall	1.01	0.009	0.87	–
Feces, summer†	2.33	1.04	0.32	81.9

† Results are based on dry weight basis.

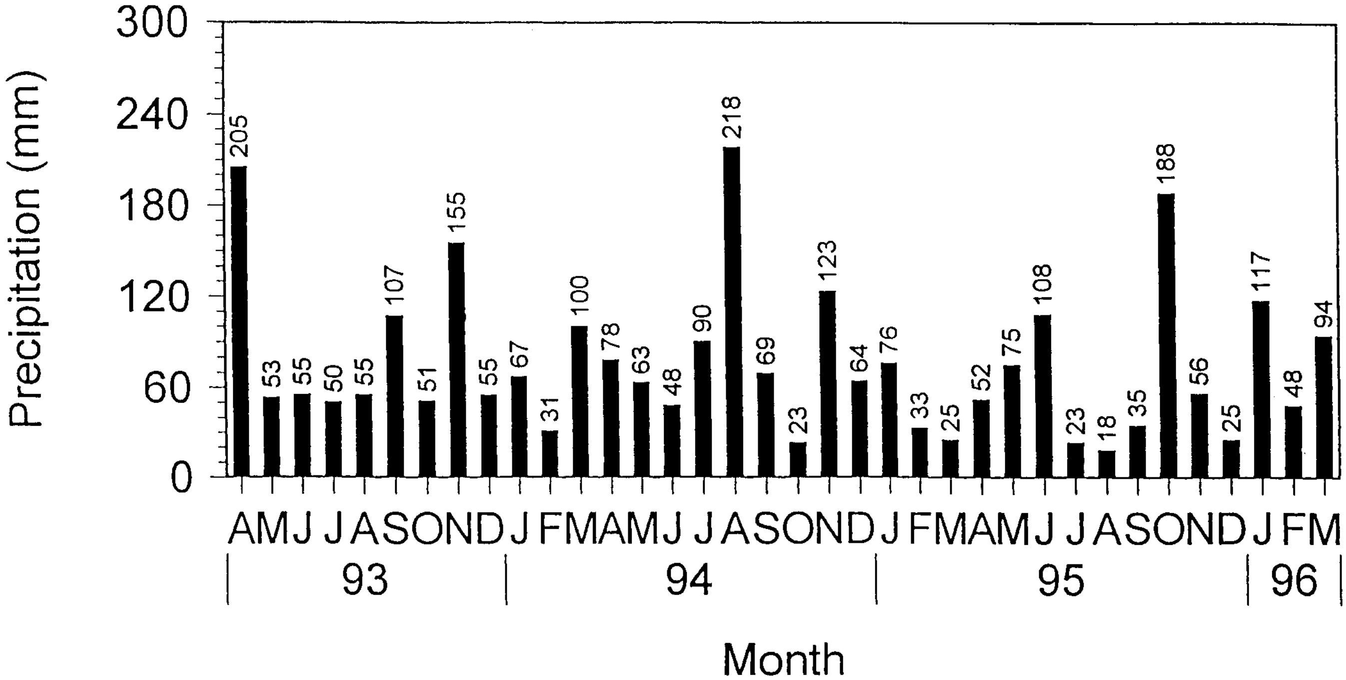

Fig. 2. Monthly precipitation distribution.

driving variables and inputs for the SOIL model. The water and heat flow are based on two coupled differential equations derived from Darcy's and Fourier's equations, respectively, in a vertical soil profile. Soil hydraulic properties are described by the water retention characteristics curve in the form proposed by Brooks and Corey (1964), and the unsaturated hydraulic conductivity function based on Mualem's equation (Mualem, 1976).

The SOILN model simulates the daily N and C fluxes in agricultural systems, including plant growth and N uptake. The SOIL model supplies driving variables in SOILN. The model simulates N transformations as functions of soil water content and temperature, N leaching, and plant N uptake. The soil profile is divided into layers, each of which includes inorganic and organic pools. The inorganic N pools are nitrate and ammonium. The organic N pools are divided into a litter pool consisting of undecomposed materials, a humus pool consisting of stabilized decomposed material, and a manure-derived feces pool. The N dynamics of litter and feces depend on C dynamics of the pools. Further information regarding the SOIL–SOILN model, equations, and estimation of the parameters is given in Jansson (1991) and Eckersten et al. (1996).

Modeling and Model Execution

The SOIL–SOILN model requires a manageable input of soil physical, hydraulic, and chemical characteristics for each layer or horizon; soil nitrogen transformation components and their rate constants; weather and hydrological data; and crop and management information for the site. The climate data for each year were collected at a weather station established at the site. Monthly precipitation distribution for 1993 through 1996 is given in Fig. 2.

The SOIL–SOILN was run for each of the three years with simulation periods beginning in May 1993 and continuing through March 1994 for 1993–1994 and in April of one year and continuing through March of the following year for both 1994–1995 and 1995–1996. The simulated water flow and mass of NO_3–N leached were compared with the mean of replicated field data collected from the lysimeters for each year.

The SOIL–SOILN model was previously calibrated using 1989–1990 water drainage and nitrate leaching data for the control, N-fertilized, and manure treatments under corn fields on Hagerstown soils in central Pennsylvania (Jabro et al., 1999). The calibrated soil and N transformation rate constant parameters developed from the N-fertilized and manure treatments from a previous study on similar soils under corn were used to test the model for 1993–1994, 1994–1995, and 1995–1996 water drainage fluxes and nitrate losses from orchardgrass pasture.

The N transformation rate constants determined through the calibration process for the corn N-fertilized (inorganic fertilizer) treatments were applied to control and urine treatments, while the rate constants for manured corn were applied to the feces treatment (Table 3).

Statistical Model Simulative Performance

Evaluation of the SOIL–SOILN performance and accuracy to simulate total water drainage and NO_3–N leaching during each year was assessed using two statistical procedures. Simple linear equations were generated from the regression analysis of simulated and measured monthly values of each year using the SAS regression procedure (SAS Institute, 1999). A test of the null hypothesis of an intercept of zero, a slope equal to one, and coefficient of determination (R^2) were used as a measure of degree of association between simulated and measured values. Model predictions were also assumed to be accurate if the predicted water flow and NO_3–N leached values fell within the 95% confidence intervals (two standard errors of the mean) of the measured data (Loague and Green, 1991; Smith et al., 1996).

Table 3. Input parameter values used in the SOIL–SOILN model.

Parameter description	Value
SOIL model†	
Saturated hydraulic conductivity (cm min^{-1})	0.15–0028‡
Pore size distribution index (Brooks–Corey equation)	0.19–0.12
Air-entry pressure (cm)	20–3
Residual water content (%)	3.0–2.7
SOILN model	
Humus specific mineralization rate (10^{-4} d^{-1})	2–8‡
Litter specific decomposition rate (d^{-1})	0.04
Manure specific decomposition rate (d^{-1})	0.04
Litter carbon humification fraction (d^{-1})	0.12
C to N ratio of decomposer biomass	10
C to N ratio of humified products	10
Specific nitrification rate constant (d^{-1})	0.3–0.5‡
Q10	2.0
C to N ratio of above ground residues	50
C to N ratios of roots	25
Denitrification potential rate (g N m^{-2} d^{-1})	0.06–0.1
Half saturation constant (mg L^{-1})	10
C to N ratio of manure	16
Fertilizer specific dissolution rate (d^{-1})	0.3
Efficiency of internal synthesis of microbial biomass and metabolites in manure	0.5‡

† Represents range values within soil profile.
‡ Parameters adjusted during calibration process.

RESULTS AND DISCUSSION

Water Drainage SOIL Simulations

The SOIL water drainage simulations for 1993–1994, 1994–1995, and 1995–1996 were compared with the average of the 23 lysimeters of measured water flow below the 1-m depth using two statistical measures (Table 4). The statistical results for drainage fluxes for these three years are given in Table 4. The intercept and slope of linear regression were not significantly different from zero and one, respectively. Furthermore, the model annual predictions of water flow fell within 95% confidence intervals (two standard errors of the mean of the measured values, 2SE). The statistical analyses reported in Table 4 demonstrated a good simulation of annual water drainage fluxes below the 1-m depth for all three years under orchardgrass sod using the SOIL model.

Generally, these results suggested that the SOIL model has the potential to simulate the amount of drainage losses below the 1-m depth in these three years (1993–1994, 1994–1995, and 1995–1996) under orchardgrass pasture conditions. Despite the statistical fit between the simulated and measured values, the model appeared to consistently overestimate the measured annual drainage fluxes in all three years. There is no obvious explanation for this overestimation in these three years.

Table 4. Performance of SOIL model to simulate annual water drainage.

Year	*N*†	Intercept	Slope	R^2	Cumulative water drainage: Measured (mean ± 2SE)‡	Cumulative water drainage: Simulated
					mm	mm
1993–1994	11	5.79	0.84	0.91	232 ± 73	255
1994–1995	12	1.4	1.02	0.96	243 ± 81	262
1995–1996	12	3.05	0.92	0.94	227 ± 81	244

† *N* = number of measurements.
‡ SE = standard error of the mean.

Nitrate Nitrogen SOILN Simulations

The N parameters, N transformation, and their rate constants determined through the calibration process on Hagerstown soils under corn (Jabro et al., 1999) were applied to the 1993–1994, 1994–1995, and 1995–1996 years to evaluate the accuracy of the SOILN model under pasture conditions by comparing the annual simulations with the mean of field-measured data.

The statistical results analysis in Table 5 showed that the SOILN model performed well and gave accurate annual simulations of measured nitrate leaching for all urine treatments in these three years. However, the model failed to accurately simulate NO_3–N fluxes for the two control treatments in 1993–1994 and 1995–1996, and the feces summer treatment in all three years (Table 5); simulated values fell outside the 95% confidence intervals of the measured values. The slope of the regression line was also significantly different from one for these five treatments (Table 5). The model annual simulations of NO_3–N fluxes for the feces treatment in 1994–1995 actually did fit within two standard errors of the mean (95% confidence interval of the mean), which could be attributed to a high variability (standard error = 0.46 with a mean of 0.40) among five replications in this data set.

On the other hand, the model was also able to successfully simulate annual NO_3–N leached for the control and other treatments (except the feces summer treatment in 1994–1995), as indicated by regression parameter (intercept and slope) values that were not significantly different from zero and one, respectively. Meanwhile, annual simulations of NO_3–N fluxes fell within the 95% confidence interval of the measured data (Table 5).

Despite the statistical fit between the simulated and measured values, the model showed a tendency to un-

Table 5. Performance of SOILN model in predicting annual loss of NO_3–N through leaching below the 1-m depth.

Treatment	*N*‡	Intercept	Slope	R^2	Cumulative NO_3–N leached: Measured (mean ± 2SE)	Cumulative NO_3–N leached: Simulated
					g m^{-2}	g m^{-2}
1993–1994						
Control†	11	0.04	0.40*	0.61	1.6 ± 0.36	1.0
Urine, spring	11	0.54	0.76	0.92	17.8 ± 0.8	18.2
Urine, summer	11	0.27	0.88	0.94	11.6 ± 0.9	12.1
Urine, fall	11	0.27	0.86	0.99	21.8 ± 0.85	21.6
Feces, summer†	11	0.10*	0.51*	0.64	2.16 ± 0.13	1.65
1994–1995						
Control	12	0.02	0.63	0.79	1.68 ± 0.5	1.3
Urine, spring	12	0.23	0.85	0.78	20.0 ± 5.0	19.8
Urine, summer	12	0.06	0.80	0.81	18.3 ± 4.5	15.3
Urine, fall	12	0.04	0.74	0.96	29.1 ± 17.0	22.0
Feces, summer	12	0.02	0.40*	0.34	1.8 ± 0.92	1.05
1995–1996						
Control†	12	0.01	0.52*	0.94	1.3 ± 0.28	0.74
Urine, spring	12	0.32	0.71	0.96	35.9 ± 5.3	32.3
Urine, summer	12	0.25	0.86	0.97	41.6 ± 4.3	38.6
Urine, fall	12	0.26	0.75	0.99	34.7 ± 9.2	29.0
Feces, summer†	12	0.01	0.50*	0.96	1.92 ± 0.56	1.11

* Regression coefficients are significantly different from zero and one, respectively, at the 0.5 probability level.
† Indicates that simulated values are not within the 95% confidence interval. (2SE) of the measured values (SE = standard error of the mean).
‡ *N* = number of measurements.

derestimate the annual NO_3–N leached below the rooting zone of orchardgrass in all six control and feces treatments and seven of the urine treatments (13 cases out of 15). There was no obvious reason for this model underestimation of NO_3–N losses under these conditions. These underestimations were in agreement with results found by Jabro et al. (1999) under a corn cropping system.

The cause for the differences between model predictions and measured NO_3–N leached below the 1-m depth for the control and feces treatments in these three years could have resulted from using N mineralization rate constants determined during the calibration process under corn (Jabro et al., 1999). The humus specific mineralization rate parameter appeared to greatly affect the production of N and the amount of NO_3–N leached in all treatments. This is particularly true for treatments where amendments (i.e., feces, manure, compost) are added. The other parameters that seemed to be the most sensitive for the feces or manure treatments were the specific nitrification rate and the efficiency of the internal synthesis of microbial biomass and metabolites (Table 3).

Despite these discrepancies in the results for the control and feces treatments, the SOILN model successfully provided accurate annual simulations of nitrate leaching beyond the rooting zone of orchardgrass for one control treatment and all urine treatments in these three years.

Overall, the results from this study showed that the SOILN model has the potential to predict the fate of fertilizers or feces N added to orchardgrass in relation to NO_3–N leaching losses below the 1-m depth using N parameters derived from previous experiments with corn. Therefore, this work may represent a step forward in the process of model field-testing and evaluations. Further field-testing using data from various soils, crops, management, and weather conditions are needed to generate the model's application to real field conditions.

CONCLUSIONS

The results from the statistical analyses suggest that the SOIL model was able to provide accurate annual predictions of the measured drainage water fluxes from the soil profile collected below the 1-m depth for all three years under orchardgrass pasture. Further, the results showed that the SOILN model was able to simulate annual total NO_3–N leached below the root zone of orchardgrass for 10 of 15 cases. The model did not adequately predict NO_3–N leached below the 1-m depth under all control and feces treatments. These variations in the results for the control and feces treatments seemed to be linked primarily to parameters controlling the N mineralization process in the model.

In general, these modeling results demonstrate the potential of SOILN to predict NO_3–N fluxes under pasture conditions using N transformation rate constants derived through the model calibration process from corn fields. While the results of this study may represent an improvement in the modeling of leaching of agricultural chemicals, this work is considered only one step forward in the process of model field evaluations.

Based on these modeling results and those found by Jabro et al. (1999), the N transformations algorithm probably requires modification to accommodate these type of conditions, where feces and manure amendments are applied.

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Particulate Phosphorus and Sediment in Surface Runoff and Drainflow from Clayey Soils

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ABSTRACT

Recent work has shown that a significant portion of the total loss of phosphorus (P) from agricultural soils may occur via subsurface drainflow. The aim of this study was to compare the concentrations of different P forms in surface and subsurface runoff, and to assess the potential algal availability of particulate phosphorus (PP) in runoff waters. The material consisted of 91 water-sample pairs (surface runoff vs. subsurface drainage waters) from two artificially drained clayey soils (a Typic Cryaquept and an Aeric Cryaquept) and was analyzed for total suspended solids (TSS), total phosphorus (TP), dissolved molybdate-reactive phophorus (DRP), and anion exchange resin–extractable phosphorus (AER-P). On the basis of these determinations, we calculated the concentrations of PP, desorbable particulate phosphorus (PPi), and particulate unavailable (nondesorbable) phosphorus (PUP). Some water samples and the soils were also analyzed for ^{137}Cs activity and particle-size distribution. The major P fraction in the waters studied was PP and, on average, only 7% of it was desorbable by AER. However, a mean of 47% of potentially bioavailable P (AER-P) consisted of PPi. The suspended soil material carried by drainflow contained as much PPi (47–79 mg kg^{-1}) as did the surface runoff sediment (45–82 mg kg^{-1}). The runoff sediments were enriched in clay-sized particles and ^{137}Cs by a factor of about two relative to the surface soils. Our results show that desorbable PP derived from topsoil may be as important a contributor to potentially algal-available P as DRP in both surface and subsurface runoff from clayey soils.

Since the 1970s, the point-source P load on Finnish watercourses has been decreasing due to effective wastewater treatment by municipalities and industry. As a result, the main contributor of P to surface waters is now the P load from nonpoint sources (Rekolainen, 1993). To improve the quality of surface waters, the P load from agricultural areas should be significantly reduced, since the deterioration in surface water quality is frequently manifested by bloomings of blue-green algae in lakes, streams, and the Baltic Sea.

In Finland, the production of arable crops is concentrated in the southern part of the country, an area with abundant clayey soils that, almost without exception, are artificially drained. Measures to reduce the P load from agriculture are targeted on efforts to diminish surface runoff. However, in artificially drained soils subsurface runoff is also an important pathway for water and solutes. In a 2-yr study, Paasonen-Kivekäs and Virtanen (1998) found that subsurface drainflow accounted for 45 and 57% of the annual total runoff from a field with clayey soil in southern Finland with a tile drainage system installed in 1950. Turtola (1999) reported that 8 yr after drainage improvement of a clayey soil in southwestern Finland up to 90% of the total water flow from plots plowed in autumn to a depth of 23 cm still occurred via subsurface drainage, while the respective proportion from plots under stubble cultivation (to a depth of 8 cm) was 50 to 60%. Turtola and Paajanen (1995) and Turtola (1999) also measured high P and sediment concentrations in drainflow, comparable with those in surface runoff. Recently, attention has increasingly focused on subsurface drainage flow as a contributor to the P load from agricultural fields and pastures (Øygarden et al., 1997; Laubel et al., 1999; Hooda et al., 1999).

Water quality monitoring usually only involves analyses of DRP and/or TP. When the bulk of the P in runoff comprises PP, TP is a poor predictor of the algal-available P load since algae take up only orthophosphate and desorbable PP (Ekholm, 1998). A large part of the P in surface runoff and subsurface drainflow from clayey soils is transported by suspended soil material (Turtola and Paajanen, 1995; Heathwaite and Johnes, 1996;

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Abbreviations: AER, anion exchange resin; AER-P, anion exchange resin–extractable phosphorus; DRP, dissolved molybdate-reactive phosphorus; ER, enrichment ratio; PP, particulate phosphorus; PPi, desorbable particulate phosphorus; PUP, particulate unavailable (nondesorbable) phosphorus; TP, total phosphorus; TSS, total suspended solids.